LCA Methodology

Evaluating Options in LCA: The Emergence of Conflicting Paradigms for Impact Assessment and Evaluation

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Abstract

LCA aims to help direct decisions in an environmentally sustainable direction. It indicates the environmental effects of choices and evaluates these against this background. Approaches to evaluation in LCA differ substantially, related to the way of modelling environmental effects and to the way these effects are combined into an overall judgement on alternative options. Several approaches are now operational, which are linked to different paradigms in decision making. It is shown that the choice of paradigm is quite decisive on the outcome of the analysis. Also within similar paradigms, different methods now operational may lead to different outcomes. These latter differences may be alleviated more easily than those related to paradigmatic choices, as they are partly a matter of refinement, and they partly result from legitimate differences in subjective priorities. The more basic paradigmatic differences can hardly be bridged. The practical relevancy of the subject is proven by applying different operational methods to one case, showing widely differing outcomes. The paradigm behind evaluating environmental effects is either values based, directly or through policy decisions, or economics based, as individual preferences measured in the monetary terms of willingness-to-pay. Accordingly, the different methods are "policy-oriented" or "monetary". It may be doubted if the differences between these can be overcome in standardisation.

Keywords: Decision making, LCA; evaluating options in LCA; impact assessment; LCA;LCIA; Life Cycle Assessment; Life Cycle Impact Assessment; monetary-based methods in LCA; policy-oriented methods in LCA; standardisation; sustainable development; tyres, LCA; weighting methods, LCA

1 Introduction

LCA has as an ultimate aim to contribute to sustainable development. In this process, it is to indicate effects of product related choices and to evaluate them against this background. As no clear definition of this ultimate aim exists or is agreed upon, there is a necessarily subjective component in the choice of evaluation method and content. Hence, there are a number of quite different evaluation[1] methods available which up to now have not been compared sufficiently. Comparison of the results of the methods as applied to case studies may give the impression that the weighting element is quite arbitrary and that it would be better to omit it from LCIA. However, as decisions have to be made anyway, it is useful to try more consistently to explain the differences between the various methods. In part these may be due to incompleteness, which can be overcome with further development and in part they may be due to different paradigms, regardless of the types of application. As to the last, there is a need (at least) for greater clarification.

This article offers a framework for comparing the different methods used in the policy and monetary approach. This may contribute to further development, it may lead to better specification of which method should be used in which situation, and it may provide a first step towards standardisation. First a framework for comparing the evaluation methods will be developed on the basis of which some more or less established methods used in LCA will be compared; these methods are: Ecoscarcity, EPS, Ecoindicator 95, SETAC

[1] The step called *evaluation* or *weighting across impact categories* in this paper is defined by the SETAC as *valuation* and in the ISO framework as *weighting*. These are the respective definitions:
SETAC : "*valuation* is the step in which the contributions from the different impact categories are weighted so that they can be compared among themselves" (CONSOLI et al., 1993).
ISO "*weighting* aims to rank, weight, or possibly aggregate the results of different life cycle impact assessment categories in order to arrive at the relative importance of these different results" (ISO/TC207/SC5, 1996).
The step termed *evaluation* in the ISO framework is different since it refers to the acceptability of the results and includes such tools as a completeness check, sensitivity check and consistency check.

method[2] with NOGEPA weighting set and the method developed in the ExternE project. Then the methods will be compared with reference to a case study and the methods and results discussed. Further to other studies on the comparison of weighting methods (BAUMAN et al., 1994; BOVY et al., 1994; HERTWICH et al., 1997), a monetary valuation is also combined with the same inventory using results taken from the methods developed in the ExternE project. This method, developed for comparing different systems of energy production, is based on the impact pathway approach where emissions are valued according to willingness-to-pay for the impacts. The impact pathway approach finds its origins in Cost Benefit Analysis, and has been applied in cases in which LCA can also be applied. This paradigm clearly is receiving momentum in Europe through active dissemination by the European Commission. In the US, and to a lesser extent in the UK, there already is a strong regulatory background for applying such monetary methods.

2 Methods Used for Evaluation in LCA

The evaluation of alternative options requires an empirical and a normative component. The empirical part is specified in the inventory analysis, which is presumed here as given, and in the impact assessment. In the impact assessment, the empirical component consists of scientific information of environmental effects in terms of the characterisation models chosen for the impact categories. These environmental effects, as on climate forcing, acidification and toxicity, cannot be evaluated directly but only indirectly by assessing their further effects on items directly relevant for evaluation, as on human health, environmental quality and on marketed resources and goods. These "endpoints" usually cannot be modelled adequately due to uncertainties of a scientific nature and to conditionalities. For example, the effects on climate change through sea level rise, potentially killing millions of people, may be prevented through migration and dike building. Owing to the lack of adequate scientific knowledge about the mechanisms by which the environmental impact categories affect the valued endpoints, the empirical component is uncertain. The normative component consists of information about the social preferences attached to the different impact categories. This implies a social or economic procedure. Combination of these last two components can lead to definition of a set of weighting factors for the environmental problems that indicates the relative contribution of each problem to their combined effects on the safeguard subjects: human health, value of ecosystems and availability of resources (LINDEIJER, 1996).

Technically, the various approaches to evaluation may differ in many aspects: they may be based on normalised vs. non-normalised data, on an average vs. marginal approach, on potential versus actual effects, on the point in the cause-effect chain where they are performed, or on different methods to establish collective preferences. Normalisation is defined as an optional element relating all impact scores of a functional unit to the impact scores of a reference situation (HEIJUNGS et al., 1996). All the policy-oriented methods compared are based on normalised data relating to impact categories or to total substance load. The ExternE method is not based on normalised data, its impacts being expressed in monetary terms on the basis of the preferences of those affected by these impacts. Among the different methods compared, ExternE is the only one based on a marginal approach since it looks at the incremental effects or damage of a reference plant background situation. As the effect models are still linear through the origin this choice does not yet have practical consequences. The method requires knowledge of many damage functions which, however, are available only for few substances; for this reason it is expected that the overall burden as calculated with the ExternE method in the case study will be due to a few substances only. The same approach is followed partly in the EPS method, but not in the policy methods – Ecoscarcity, SETAC with NOGEPA, Ecoindicator 95- which are based on a site-independent approach that implies the assessment of potential effects. Whether or not to include spatial and temporal aspects in the SETAC framework (HEIJUNGS et al., 1993; POTTING et al., 1994; WRISBERG et al., 1997), however, is a point of current debate within the LCA community.

Table 1 classifies several evaluation methods with reference to the point in the cause-damage chain – interventions, impact indicators, damage to valued endpoints – where they are performed and to the choice of the method used to establish collective preferences. The cause-damage chain starts from the interventions (emissions, extraction, land use) contributing to certain environmental impacts (e.g. global warming); these impacts are quantified by means of certain impact indicators (climate-forcing) that are responsible for certain damages (value of lost land due to sea-level rise) to certain receptors. The evaluation methods can be performed at any level in the cause-effect chain: at the substance level no information on the impact category involved will be supplied; at the impact categories level more knowledge will be available; at the damage level a much better picture of damages will be available, but there will often be insufficient scientific knowledge to fully operationalise this approach.

There are five main approaches in LCA evaluation for establishing collective preferences at different points of the cause-effect chain: individual preferences, collective revealed, collective subjective, policy criteria and objective criteria (HUPPES et al., 1997).

1. In the individual preferences methods, the economic mechanisms set the weights through the two tools of contingent valuation: equivalent variation (willingness-to-pay), which states how much each person involved would be willing to pay to prevent a certain deterioration of the environment, and compensating variation (willingness-

[2] SETAC characterisation as operationalised in the Guide and Background (HEIJUNGS et al., 1992).

to-accept), which states how much the person involved would want to receive to consent to this deterioration. In this approach, weighting appears feasible only at the level of the damage categories or "endpoints". As examples of this approach, the EPS, which is based not only on economic mechanisms, and the ExternE project are considered. All other methods ultimately are partly or fully value based.

2. In the collective, revealed preferences approach the weights are derived from government decisions; they are the highest costs still acceptable for the reduction of one unit of environmental problem. The weighting can take place at the level of the interventions, the impact categories, the damage types or the safeguard subjects. A variety of unpublished government reports identifying priority options for environmental investments form examples of this approach at the impact categories level.

3. In the collective subjective preferences, or societal approach, the weighting can be performed at the level of the impact categories, the damage categories, or the safeguard subjects, e.g. by a panel (VOLKWEIN et al., 1996) representative of the society as a whole. In this study, the NOGEPA panel is considered as an example.

4. In the policy criteria approach, the weights are derived from government statements on emission reduction targets considered to constitute a no-effect level. This approach is usually based on the distance-to-target method or ratio-to-target method; the weighting can take place at the level of the interventions, the impact categories, the damage categories or the safeguard subjects. The Ecoscarcity and Ecoindicator 95 methods are examples of this approach.

5. In the objective criteria method, certain sustainability levels are identified for each impact category and the distances to these sustainability factors are taken to represent the weights (GUINÉE, 1995). In order to arrive at a societal evaluation, a "cross-category factor" representing societal preferences must still be applied. Without this factor, the objective criteria method does not represent a full method.

In the following paragraphs, the operational evaluation methods shown in **Table 1** are briefly reviewed. Since the first three methods described belong to familiar policy approaches, a longer description of the remaining two monetary approaches is presented.

2.1 The "Ecoscarcity" approach

The Ecoscarcity approach (AHBE et al., 1990) relates the concept of economic scarcity (relation between supply and demand) to ecological topics by defining ecological scarcity as the relation between the critical load of a substance and the actual load of anthropogenic emissions of that substance. In this method, the different interventions are weighted against one another directly; the weighting takes place at the first level of the cause-effect chain. The method is based on a distance-to-target (or ratio-to-target) between the critical and the actual flow of a substance. Both the critical and the actual flows are derived from the national conditions: the critical flow is the annual load limit set by national environmental laws and the actual flow is the total magnitude of an intervention in a specific area during a certain period. The Ecoscarcity approach has been adapted to the SETAC framework by translating ecoscarcity at the intervention levels into ecoscarcity at the effect level (MÜLLER-WENK, 1994).

The main advantage of the Ecoscarcity is its simplicity and ease of use.

The main disadvantages of this method are that:

- the results are dependent on political priorities;
- the political target and the critical flows are site-specific; this implies that there will be different ecofactors for every country, and not a formalised set of weighting factors independent of the site (TUKKER, 1994).

Table 1: Classification of evaluation methods

	(1) emissions/ extractions/ land use	(2) impact indicators	(3) damage categories	(4) safeguard subjects
individual preferences			ExternE	EPS
collective revealed		Unpublished government reports		
collective subjective		- SETAC - NOGEPA		
policy criteria	Ecoscarcity	- Ecoindicator 95 - new Ecoscarcity		
objective criteria		GUINÉE		

In this study, the Norwegian ecofactors have been used solely because they cover more environmental interventions.

2.2 The "NOGEPA panel" approach

The NOGEPA (Netherlands Oil and Gas Exploration and Production Association) panel method belongs to the collective subjective preferences approach, and since the panel establishes scores for different impact categories, its position in the cause-effect chain is at the impact categories level. The NOGEPA panel, in the context of implementing environmental policy in its sector, has developed a weighting set for environmental effects in order to identify the environmental cost-effectiveness of investment options as a starting point for prioritarisation. Because the effects as assessed by the panel are mainly in line with the effects contemplated by the SETAC LCA guidelines, the NOGEPA weighting set can be used in LCA case studies performed with the SETAC method. The SETAC LCA method, for which readers are referred to Consoli, Heijungs and ISO 14040, is not described here. In the NOGEPA project, a panel of 26 people from ministries, companies and universities was established to make a quantitative pronouncement on the relative relevance of the environmental effects associated with Dutch atmospheric emissions. The time horizon for effects was defined as 20 to 50 years, under the assumption that the level of emissions would remain the same; the spatial area was not limited, a global scale being assumed for all the effects. In the NOGEPA project, the panel weighting set represents the "starting point" for prioritising investment options. It can readily be used for weighting across impact categories in other LCA case studies. Its limitation is that it is incomplete, because it only considers emissions to the atmosphere and certain other categories of effects.

In this study, the following effects have been assessed: abiotic resource depletion, global warming, ozone depletion, acidification, nutrification, photochemical oxidant formation, human toxicity and ecotoxicity. In the NOGEPA procedure, no weight was established for abiotic depletion or ecotoxicity; here, these weights have been estimated assigning these impacts the same importance as all other effects. Normalisation of the impact categories score has been carried out using the most recently available normalisation data for Europe (Directoraat-Generaal Rijkswaterstaat, 1997).

2.3 The "Ecoindicator 95" approach

In the Ecoindicator 95 method (Goedkoop, 1995), the different impact categories are weighted relative to one another and then summed to form an overall environmental index. Ecoindicator 95 is fundamentally a distance-to-target method; the weights are based on the ratio between the actual annual contribution to a problem and the no-effect contribution to that problem. The targets are set according to the following criteria: one extra death per million inhabitants per year; health complaints as a result of a smog episode; five per cent ecosystem impairment.

This method derives weighting factors for the different impact categories that are generally in line with the SETAC guidelines framework; it can be readily used to carry out LCA case studies even if there are some differences in the definition of the impact categories[3] and in the characterisation and normalisation factors. The main disadvantages of this method are:

- there is no normative information because the "cross-category factor" is set arbitrarily at 1;
- resource depletion is not assessed.

The new version of this method, which is currently under development, seems to have adopted an approach more similar to the ExternE project, based on the damage functions but without indicating the relative importance of an impact category. It adopts the concepts of DALY's, – Disability Adjusted Life Years – (Murray et al., 1996) and PAF's -Potentially Affected Fraction of species – (Klepper et al., 1997): the former is a health indicator based on epidemiological data that measures the total amount of ill health due to disability and premature death attributable to specific diseases and injuries; the latter is an index that calculates the fraction of species exposed to values above the NOEC for a given environmental concentration of a substance.

In this study, the method as proposed by the designers of the Ecoindicator 95 method has been used by adopting its characterisation and European-based normalisation factors; in contrast to the NOGEPA method, in this case no estimate has been made of the missing weights (abiotic depletion and ecotoxicity). This difference in treatment is due to the fact that the NOGEPA project was concerned only with the problem associated with atmospheric emissions; this implies that there is nothing wrong with adding estimated weights for two impact categories reckoned to be important. By contrast, the Ecoindicator 95 method was designed as a weighting method for environmental effects that damage ecosystems or human health; this means that it is precisely these effects that are considered relevant by the author.

2.4 The "EPS" method

In the EPS system (Steen et al., 1993), the environmental interventions are evaluated according to the effect they have on the following safeguard subjects: human health, biological diversity, production, resources and aesthetic values. The set of default factors applied directly to the inventory is based on a procedure carried out at both the impact categories

[3] The depletion of resources is not considered and human toxicity is replaced by the categories summer smog, carcinogenic substances, heavy metals and winter smog.

and the safeguard level. The emissions are related to the safeguard subjects by going through the SETAC procedure of classification and characterisation combined with certain correction factors, e.g. for the extent, intensity and duration of the effect. Valuation of the interventions in monetary terms is then performed on the basis of a willingness-to-pay for restoration of the safeguard subjects and the availability of sustainable substitute processes. Loss of human health, biodiversity and aesthetic values are monetarised using contingent valuation to establish the price society is willing to pay to avoid these damages. Resource depletion is monetarised by looking at the future environmental costs of substitute processes (environmental costs that must be incurred to recover the minerals or the fossil fuels from other rocks or sources with the help of a biotic energy source). Production losses are monetarised by measuring estimated reductions in agricultural yields. The EPS environmental factors are expressed in ELU (Environmental Load Units) per kg. This unit is assumed to be equal to the ECU but is termed differently because it does not represent a real market value but a common value that can be used to compare different measures.

The EPS method focuses more on resource depletion than on emissions.

2.5 The "ExternE" method

The main objective of the ExternE project (EC, 1995) is to develop a methodology to assess the external costs of the energy sector. The external costs or negative externalities are defined in economic terms as "the costs which arise when the social or economic activities of one group of people have an impact on another, and when the first group fail to fully account for their impacts". The externalities derived from energy production (or, more generally, from any productive cycle) with respect to the welfare of society are the damages to human health, ecosystems and materials whose costs are not reflected in the market price of the energy. The ExternE approach involves quantification of the environmental and health impacts of certain fuels – coal, lignite, oil, gas, nuclear and renewables – and their relative external costs during the different phases of the life cycle.

The impacts identified, quantified and partly monetarised are the following: occupational and public accidents; effects of atmospheric pollution on human health, crops, forests, freshwater fisheries and unmanaged ecosystems; effects of global warming by greenhouse gases; and effects of noise. Quantification of the damage caused to the five receptors – human health, buildings and materials, crops, forests and fresh waters – is performed using the damage function or impact pathway approach. The damage function approach places great emphasis on the relationships between natural and economic sciences in revealing the sequence of changes that result in damage (Cantor et al., 1992). In order to apply this method, specific reference plant technologies and means of transport have to be considered. This means that some of the damages estimated are dependent on the reference site chosen. The method adopts a marginal approach as it looks at the incremental effects or damages of the single reference plant being considered relative to the background conditions. It is divided into four steps:

1. inventory of the environmental burdens occurring at specific sites;
2. calculation of the increased pollutant concentrations in the regions affected;
3. calculation of the physical impacts;
4. economic valuation of the damage.

The inventory of the environmental burdens derived from the different sites and transport steps of the fuel cycle is quite straightforward and is very similar to the set-up of the LCA inventory. The main difference is that the LCA inventory and all the other methods described, except EPS, deal with potential impact. They do not specify how the interventions occur in real time and space dimensions, while the ExternE inventory relates every intervention to a specific site in order to apply the impact pathway methodology. Atmospheric transport flows are estimated using models of atmospheric dispersion and chemistry to assess the fate of the pollutants over a distance of several hundreds of kilometres. The physical impacts are estimated using dose-response functions which may be linear or complex.

The final stage is economic valuation which is accomplished using three tools: market values, indirect valuation, for example in terms of averting behaviour[4], replacements costs[5], travel costs[6] or hedonic pricing[7] and contingent valuation. The first technique can be used when the receptors of the damage are commodities which have a real market, for example crops, timber or buildings. The different indirect valuation prices are used to value impacts on public goods, such as recreational sites, by observing people's behaviour in a real market. Contingent valuation is used to value damages having no connection with real markets, such as biodiversity. Because of the current controversy discussion regarding the discounting of the monetary values of environmental and health impacts, three different rates, 0%, 3% and 10%, have been used where necessary.

[4] This method is based on the concept of perfect substitutability: what people pay to have substitutes for environmental change, for example, bottled water rather than tap water.

[5] This technique uses the costs of replacing a damaged asset to its original state.

[6] In this technique, what people pay in terms of travel and time costs can be used to set a value on natural resources such as a lake.

[7] This method considers the real market for commodities whose prices are affected by environmental factors; a typical example is the different price of two houses that are identical in all aspects, except for one being located near an airport.

The results of the ExternE project can be used in the LCA evaluation to obtain a monetary value for some pollutants. Two main problems arise in applying them to LCA case studies. First, monetary values are operational for only few substances, such as dust, SO_2, and NO_x, so the overall results would only be related to these; although it is possible to use monetary values for other pollutants taken from other studies so that there will always be a variety of different values available (LEE et al., 1996). Second, the results are site-specific. Environmental differences in the background of the sites and socio-economic differences in the population structures can seriously affect the results: the first by affecting the magnitude of the damages, the second by affecting the willingness-to-pay. Meta-analysis can be undertaken to transfer the result of one site-specific study to another, but the results would simply be estimates.

In this study, two different cases have been considered using the ExternE approach: one based only on the ExternE results, the other on the ExternE results plus some additional values. In the first case, the pollutant values presented in the European Commission report (EC, 1996) (averages values for the different countries considered) have been adopted; only dust, SO_2 and NO_x are assessed. The second case has a set in which, additional to the ExternE results, there are some values for greenhouse gases as presented in (FANKHAUSER, 1994; CRAIGHILL et al., 1996). Other studies reveal different monetary values for the same pollutants (e.g. HORMANDINGER, 1995).

3 Evaluation Results for the Case of Car Tyres

Theoretical differences may be profound but without real consequences in practice. To test the practical consequences of a choice of paradigms and methods, we apply a number of operational methods on a case. Any more or less complex case could have been chosen. We took one from the practical work of the first author, a cradle-to-gate inventory of one tonne of car tyres (NICOLETTI et al., 1998). This inventory, reported in **Table 2** in a summarised manner, considers the resource consumption and the environmental burdens associated with acquisition of the main materials required for manufacturing one tonne of car tyres and with the actual production phase; it is a cradle-to-gate analysis. The functional unit is one tonne of car tyres. The product system is the production of one tonne of car tyres from the extraction of raw materials through production. The scope of the analysis is to identify the "hot spots" of the system, both at level of materials and of substances and to identify options for improvement.

The most important processes not included in the inventory because of lack of data are those associated with the production and use of fertilisers and pesticides in natural rubber production. The data used are average European data and, in some cases, are based on unpublished environmental information supplied by Italian companies. Some substances, in particular from the latter data source, are highly aggregated, without much specification of their chemical nature as required in LCA. In order to solve this problem, two constructed inventory tables have been based on the "best" (least toxic substance in the group) and the "worst" (most toxic substance in the group) factors, as defined in the Guide[8]. In the best case, for example, it follows that all unknown metals in air have been assumed to be as low-toxic as zinc, which has a characterisation factor for human toxicity of 0.033, while in the worst case the same quantity of unknown metals in air has been assumed to be as toxic as chromium (VI), with a characterisation factor for the same impact category of 47,000, more than six orders of magnitude larger.

Once the "best" and the "worst" constructed inventory tables have been set up, six different evaluation methods are applied to each case: Ecoscarcity, EPS, SETAC method with NOGEPA weighting set, Ecoindicator 95, ExternE and ExternE with some added values. The differences among the various single-step or multi-step evaluation methods are presented at three levels:

1. differences in the inventory (best versus worst);
2. differences at the materials level (steel versus all the other materials that together constitute the tyre);
3. differences at the level of substances/resources (a subdivision for all the main substances considered).

These three levels will now be considered in the following sections.

3.1 Inventory level

In **Table 3** and **Figure 1** the two inventory tables are reported on the same scale.

EPS and Ecoscarcity show almost the same characteristics, because both lack of factors for some of the unknown substances. NOGEPA shows a very large difference between the two inventories. This is due mainly to the high characterisation factor of chromium (VI) in the worst case, compared to the low zinc characterisation factor in the best case. Ecoindicator 95 has a smaller range in the characterisation factors for the heavy metals than those mentioned in (HEIJUNGS et al., 1992) which leads to a smaller difference between the two cases. The two ExternE approaches show exactly the same figures in both cases because of the lack of values for the unknown substances.

[8] The best and worst cases have been drawn up for six categories of generic substances: VOC, aldehydes and metals in air, other nitrogens, hydrocarbons and metals in water.

Table 2: Cradle-to-gate inventory of one tonne of car tyres (all values, except those indicated are expressed in kg)

Raw materials	Natural rubber	SBR	Poly butadiene	Butyl rubber	Carbon black	Rayon	Nylon	PET	Steel	Sulphur	Zinc oxide	Stearic acid	Tyre manu facturing	Total
Land use (Ha)	1.38E-1													1.38E-1
Fossil fuels (MJ)	4.74E+3	1.93E+4	3.38E+3	3.26E+3	2.89E+4	1.32E+3	2.10E+3	8.17E+2	3.30E+3	1.50E+2	7.65E+2	5.50E+2	1.63E+4	8.48E+4
Iron ore									1.65E+2					1.65E+2
Limestone									3.30E+1					3.30E+1
Coal (coke manufacturing)									8.20E+1					8.20E+1
Zinc ore											2.79E+2			2.79E+2
Sodium chloride							3.00E+1							3.00E+1
Pulp						2.20E+1								2.20E+1
Air emissions														
Dust	1.26E-1	3.43E-1	9.20E-2	3.90E-2	1.48E+0	1.93E-1	3.30E-2	3.50E-2	1.23E+0	1.00E-3	1.63E-3	1.18E-3	3.08E-1	3.88E+0
CO	5.88E-2	1.49E-1	5.60E-2	3.22E-2	8.79E-1	6.26E-2	5.04E-2	1.80E-1	7.52E+0	2.76E-4	2.84E-3	1.62E-3	3.14E-1	9.30E+0
CO_2	3.68E+2	3.96E+2	5.20E+1	7.84E+1	1.51E+2	3.50E+1	7.50E+1	2.22E+1	7.99E+1	2.98E+0	9.90E+0	5.58E+0	1.01E+3	2.29E+3
SO_x	3.05E+0	3.18E+0	5.20E-1	3.24E-1	1.38E+1	8.56E-1	2.59E-1	2.30E-1	2.38E+0	3.15E-2	6.48E-2	1.50E-2	4.67E+0	2.94E+1
NO_x	8.52E-1	1.87E+0	3.90E-1	3.06E-1	2.50E+0	1.15E-1	4.66E-1	1.43E-1	4.42E-1	6.57E-3	2.84E-2	1.47E-2	2.00E+0	9.13E+0
N_2O	2.84E-1	6.24E-1	1.30E-1	1.02E-1	8.34E-1	3.83E-2	1.55E-1	4.75E-2	1.47E-1	2.19E-3	9.48E-3	4.89E-3	6.65E-1	3.04E+0
TVOC (COV, *aldehydes, others*)	5.42E-1	3.73E+0	8.00E-1	3.60E-1	2.22E+0	1.10E-1	2.76E-1	4.83E-1	1.96E+0	2.32E-3	1.37E-2	1.03E-2	2.88E+0	1.34E+1
TVIC (HCl, NH_3, HF, H_2SO_4)	6.75E-4	3.51E-3	3.20E-3	5.37E-4	3.29E-3	2.18E-2	8.02E-4	1.00E-3	5.54E-4	4.00E-6	1.14E-3	2.55E-5	3.30E-3	3.98E-2
Sulphurised compounds (H_2S, CS_2, COS)		4.25E-4	4.00E-5	3.20E-5	5.40E-1	4.81E+0	2.40E-5		1.06E-2					5.36E+0
Metals	8.00E-5	7.43E-4	2.00E-4	3.21E-5	6.21E-4	2.44E-5	8.33E-6	1.00E-4	2.20E-3	7.00E-7	1.67E-3	8.10E-8	2.54E-2	3.11E-2
Water emissions														
BOD_5	4.83E-4	9.55E-3	1.60E-2	1.31E-3	2.30E-3	4.18E-2	3.00E-1	1.00E-2	5.06E-2	3.50E-6	1.35E-6	1.88E-6	2.26E-2	4.55E-1
COD	1.05E-3	1.72E-1	8.00E-2	6.51E-3	4.59E-3	1.36E+0	4.52E-1	3.10E-2	2.64E-6	7.00E-6	4.05E-6	5.52E-6	9.68E-2	2.20E+0
Suspended solids	4.88E-3	7.18E-2	1.20E-2	7.40E-3	3.55E-2	1.89E-2	7.48E-2	5.50E-3	1.33E-2	4.20E-5	2.41E-4	1.14E-4	1.10E-2	2.55E-1
Dissolved solids	1.75E-1	9.25E-2	1.60E-2	7.47E-2	6.99E-3	5.50E-3	5.65E-2	5.60E-3	1.38E-2		3.00E-3	3.90E-3	1.28E+0	1.73E+0
Organic compounds	3.97E-3	6.83E-2	3.52E-2	7.44E-3	2.13E-2	2.09E-1	4.23E-3	1.34E-1	5.60E-4	1.40E-5	4.00E-4	2.00E-4	3.42E-2	5.19E-1
Inorganic compounds	8.46E-4	2.17E-2	n.d.	2.10E-3	4.59E-3	3.88E-1	1.23E+0	7.50E-3	n.d.	7.00E-6	5.58E-1	2.49E-4	3.58E-2	2.25E+0
Metals	4.06E-4	1.02E-1	1.24E-2	9.75E-3	3.29E-3	8.49E-4	3.12E-3	1.10E-3	1.58E-2	3.50E-6	7.12E-2	3.60E-5	1.53E-3	2.22E-1
Other nitrogen (NH_4^+, NO_3)	2.66E-4	7.70E-3	3.23E-2	6.51E-4	1.38E-3	7.30E-6	2.77E-3		2.98E-2	2.10E-6	2.69E-5	5.76E-7	5.28E-2	1.28E-1
Solid wastes(*)														
Non-hazardous	2.53E+0	3.98E+0	1.58E+0	1.06E+0	2.85E+0	1.26E+0	1.87E+0	4.09E-1	5.74E+2	3.63E-3	8.42E+0	5.46E-2	1.70E+2	7.67E+2
Hazardous	n.a.	1.97E-2	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	7.20E-1	n.a.	n.a.	n.a.	5.00E+0	5.74E+0

(*) According to the new Italian Regulation

Table 3: Evaluations of best and worst inventory (percentage)

	EPS		ECOSC.		NOGEPA		ECO95		ExternE		ExternE+	
	worst	best	worst	best	worst	best	worst	best	worst	best	worst	best
Raw materials												
zinc ore	23.8	23.8	-	-	0.4	0.4	-	-	-	-	-	-
oil	32.0	32.0	4.0	4.0	0.0	0.0	-	-	-	-	-	-
gas	16.0	16.0	2.4	2.4	0.0	0.0	-	-	-	-	-	-
coal	5.8	5.8	1.9	1.9	-	-	-	-	-	-	-	-
Air emissions												
dust	0.0	0.0	-	-	-	-	1.2	1.2	13.3	13.3	12.8	12.8
CO_2	16.3	16.3	9.7	9.7	1.7	1.7	1.6	1.6	-	-	4.1	4.1
N_2O	1.7	1.7	16.9	16.9	0.6	0.6	0.6	0.6	-	-	0.0	0.0
NO_x	0.2	0.2	7.9	7.9	0.7	0.7	3.1	3.1	15.9	15.9	15.2	15.2
SO_2	0.2	0.2	28.1	28.1	2.7	2.7	20.6	20.6	70.8	70.8	67.9	67.9
metals	0.6	0.2	16.3	1.2	54.0	0.0	54.0	10.2	-	-	-	-
voc	1.0	1.0	11.5	11.5	2.9	1.8	3.4	5.4	-	-	-	-
Water emissions												
metals		-	-	-	36.7	0.3	15.5	0.0	-	-	-	-
others general	2.3	2.3	1.3	1.2	0.2	0.1	0.1	0.1	-	-	0.0	0.0
TOTAL	100.0	99.6	100.0	84.8	100.0	8.4	100.0	42.8	100.0	100.0	100.0	100.0

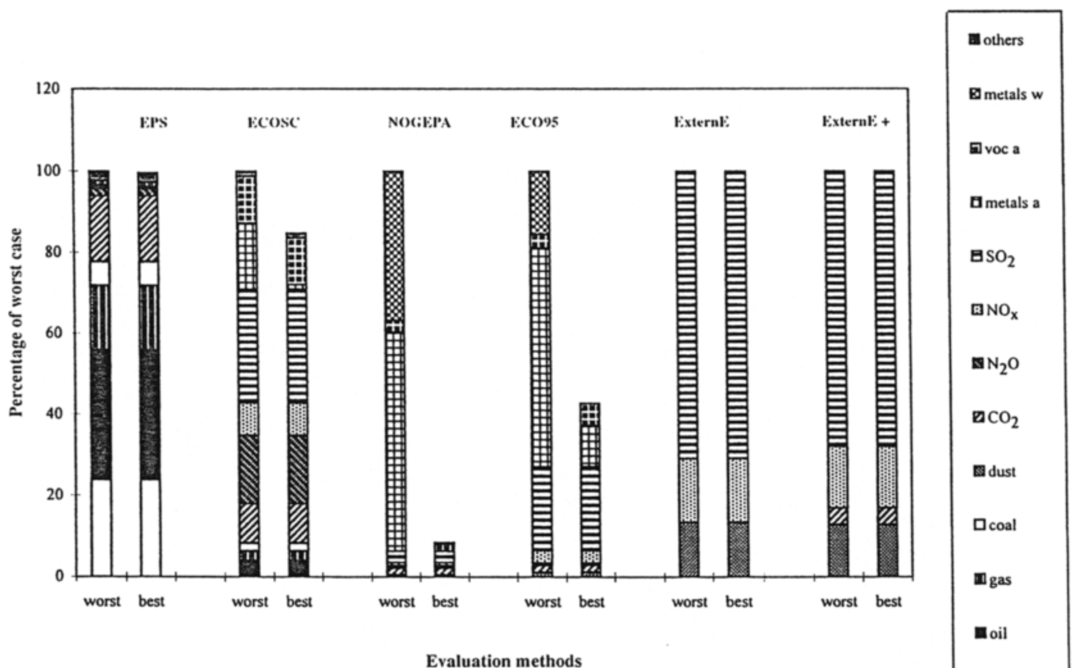

Fig. 1: Evaluations of best and worst inventory

3.2 Materials level

The material composition of the car tyre considered and the materials assessed (more than 99% of total tyre weight) are shown in **Table 4**.

In **Table 5**, the scores per kilogram of the constituent tyre materials have been normalised to the score of 1 kg of steel. In this way, the different evaluation methods are not applied to the functional unit, but to a single kg of material. The results show extremely high figures for zinc oxide; this is due to a depletion of zinc ore in the EPS method and to the emissions of metals to both air and water in the NOGEPA method.

Among the cords of the tyre, nylon 6.6 shows high scores, especially in the Ecoscarcity, EPS and NOGEPA methods,

Table 4: Materials composition of one tonne of car tyres (NOTARNICOLA, 1996)

Materials	kg/t tyre
Natural rubber	160
SBR	235
Polybutadiene	40
Butyl rubber	30
Rayon	20
Nylon	10
Polyester	10
Steel	110
Carbon black	270
Sulphur	10
Zinc oxide	15
Stearic acid	10
Other materials	80
Total	**1000**

because of its high energy content and the relatively large number of intermediates used in its manufacture. Carbon black shows high scores, especially in the Ecoscarcity method and the two ExternE methods, owing respectively to emissions of SO_2 and dust, whose characterisation factors are relevant in the ExternE method. Among the different types of rubber, which constitute almost 50% of the tyre weight, natural rubber seems to have the best score, but this result is undoubtedly due to the lack of data on fertiliser and pesticide production and use in the cultivation phase. The synthetic rubbers appear to be evaluated at more or less the same level in the various methods, but for different reasons.

Figure 2, in which the data of **Table 5** are reported graphically, shows the high range of the EPS and NOGEPA methods, due to the scores for zinc oxide.

Table 5: Scores for kg of materials normalised to kg of steel, in the best case

	ECOSC.	EPS	NOGEPA	ECO95	ExternE	ExternE+
Steel	1.0	1.0	1.0	1.0	1.0	1.0
NR	0.8	0.7	0.8	0.5	0.6	0.6
SBR	1.1	1.3	1.1	0.6	0.5	0.6
Polybutadiene	1.2	1.2	1.2	0.7	0.6	0.6
Butyl rubber	1.2	1.7	1.1	0.6	0.5	0.5
Carbon black	1.6	1.4	1.3	1.3	1.6	1.6
Rayon	1.7	1.1	1.4	1.1	1.5	1.5
Nylon	4.2	3.7	3.3	1.6	1.6	1.7
PET	1.9	1.4	2.3	1.3	0.9	1.0
Sulphur	0.6	0.2	0.6	0.7	0.8	0.8
Zinc oxide	0.4	30.7	6.4	1.2	0.1	0.2
Stearic acid	0.2	0.7	0.2	0.1	0.1	0.1

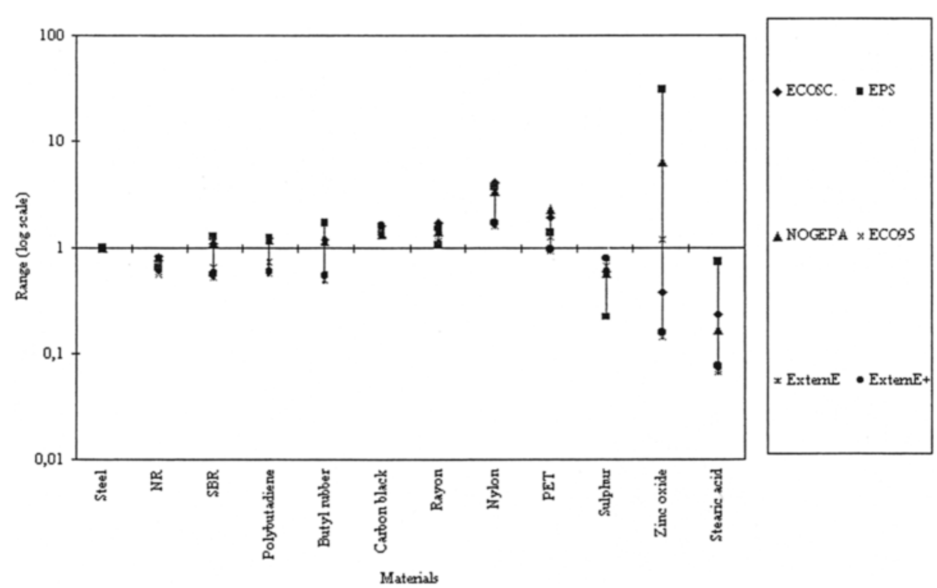

Fig. 2: Range of values for the tyre materials in different methods

Table 6: Contributions of substances to the overall result in the different methods (percentage), best case

	EPS	ECOSC.	NOGEPA	ECO 95	ExternE	ExternE +
Raw materials						
zinc ore	23.9	-	5.1	-	-	-
oil	32.2	4.7	0.4	-	-	-
gas	16.1	2.8	0.4	-	-	-
coal	5.9	2.3	-	-	-	-
Air emissions						
dust	-	-	-	2.7	13.3	12.8
CO_2	16.3	11.4	19.9	3.8	-	4.1
N_2O	1.7	20.0	7.1	1.4	-	0.0
NO_x	0.2	9.4	8.8	7.2	15.9	15.2
SO_2	0.2	33.2	32.6	48.1	70.8	67.9
zinc	0.2	1.4	0.0	23.9	-	-
styrene	1.0	13.5	21.0	12.7	-	-
Water emissions						
zinc	-	-	3.9	-	-	-
others (general)	2.3	1.5	0.8	0.1	-	0.0
TOTAL	100.0	100.0	100.0	100.0	100.0	100.0

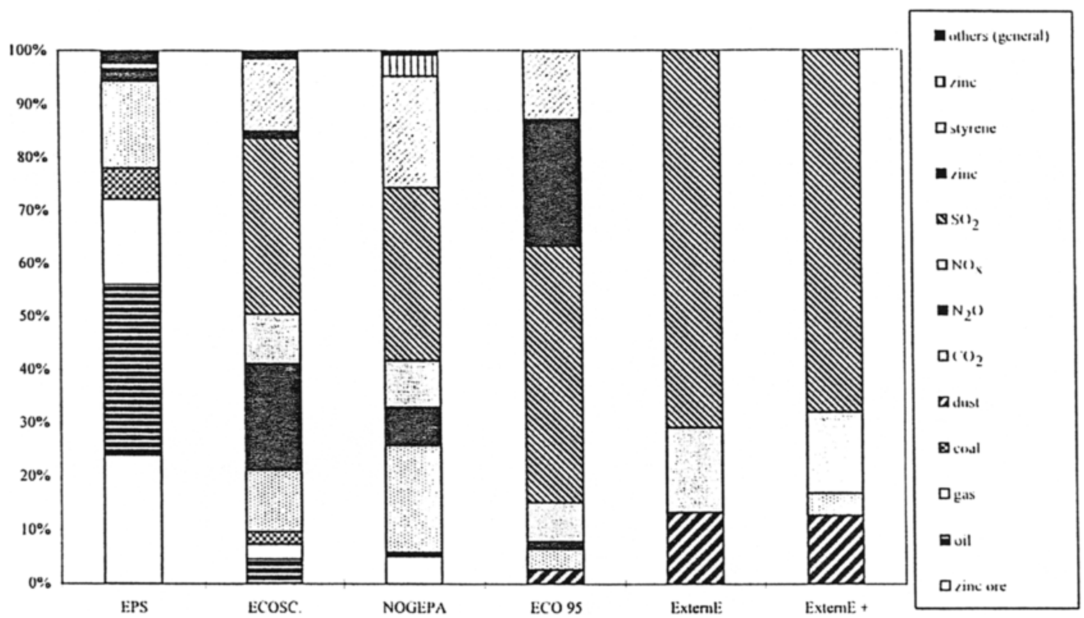

Fig. 3: Contributions of substances in the best case

3.3 Substance level

In Table 6 and Figure 3, the scores for each substance are presented as a percentage on the same scale.

From Figure 3 it can be derived that the depletion of fossil fuels together with the depletion of zinc ore in the EPS method[9], contributes to about 80% of the overall result. This is due to the high EPS factors for resource depletion. The depletion of zinc ore is relevant only for the production of zinc oxide. This result becomes much more interesting if it is observed that the zinc oxide has a share of only 1.5% in the total tyre weight. Moreover, some pollutants such as NO_x, SO_2 and styrene which demonstrate a substantial contribution in other methods are almost negligible in the EPS method, owing to the low factors assigned to them in this method.

The Ecoscarcity, NOGEPA and Ecoindicator methods show almost the same characteristics: a very high contribution of

[9] This is valid for both the best and the worst case.

SO_2, significant contributions of NO_x and a zero contribution of dust, due to the lack of a characterisation factor. The Ecoscarcity method shows no contribution for zinc ore, because there are no factors for mineral depletion, an average, high contribution of fossil fuels and a very high N_2O contribution because of the high Norwegian N_2O factor. The NOGEPA method has the highest CO_2 contribution owing to the weighting factor for global warming which is more than the double that of the others. Moreover, it has a high score for zinc ore depletion, owing to the scarcity of this mineral, and a high score for styrene due to the relatively high quantity of this substance in the inventory.

If the results of this method were to be compared with those of an equivalent method in which all the weights were set to one, the greatest differences would be in the contributions of styrene and CO_2. The NOGEPA method would register CO_2 contributions of over 10% and a styrene contribution of less than 10%.

The Ecoindicator 95 method has no value for resource depletion because this impact category is not assessed in the method, although it is affected by a very high contribution of SO_2 and of zinc in air. The two ExternE methods have the highest share of SO_2, a high contribution of dust, and an average, high contribution of NOx.

In the worst case, the EPS and the Ecoscarcity methods show almost identical results to those in the best case; this is due to the lack of factors for various aldehydes, various VOC in air and various metals in water. NOGEPA and Ecoindicator 95 show very different results from the previous case: the chromium (VI) emissions in air and water contribute respectively to about 70 and 80% of the overall result because of the very high characterisation factor of this metal. Ecoindicator 95 shows a lower contribution of chromium (VI) in water but a higher contribution of SO_2. The two ExternE approaches yield the same results as the previous case.

4 Discussion

The ExternE approach, coming from a Cost-Benefit Analysis (CBA) background, shows quite extreme differences in outcomes as compared with the various LCA-based approaches. Within the monetary paradigm, the differences are largest as EPS focuses on resource depletion and ExternE on human health. The policy oriented methods are much closer in their results. The differences in outcome appear both at the level of the tyre as a whole, i.e. the cradle-to-gate tyre inventory, and at the materials level. The results differ by a factor of more than ten.

The differences originate, to a substantial extent, from a lack of effect data on certain substances, with different gaps being observed in different methods. All the methods investigated can be extended to cover more substances. Some of the divergences would then diminish. The LCA approaches based on current policy or current policy documents, for example the Ecoscarcity and revealed preference methods, beg the question of what is important and why. The remaining methods differ fundamentally in two respects: in the point of the effect chain where assessments are made and in their philosophical basis for evaluation.

The subjective monetarised evaluation, based on willingness-to-pay, as in ExternE and to a lesser extent EPS, requires dose-response relationships for valued endpoints, implying long effect chains. As knowledge on long effect chains is limited, these methods tend to have narrow effect chains or a limited number of dose-response relationships. The focus is therefore on inhalatory problems related to fine dust and SO_2, for which empirical relationships have been substantiated statistically. Philosophically, the evaluation is based on what people consider as important to themselves. The LCA approaches based on problem themes can take into account quite broad and relatively well-proven effect mechanisms, but only in a limited way. Willingness-to-pay concepts are not applicable to this point in the effect chain; the problem themes, at best, can be seen as representing collective goods. Philosophically, evaluation is therefore based on views about what is important to society in general, without necessarily asking the individuals ultimately concerned, as is the intention in the monetary paradigm.

Assuming agreement on inventory outcomes, choices depend on the evaluation of environmental modelling results. It may be expected that the empirical part, both that of characterisation modelling and linking those results to valued endpoints, will converge within the policy-oriented paradigm. Differences in evaluation will remain but may not be decisive for outcomes. The monetary approach, quite dominant in environmental analysis outside the realm of LCA, has a set-up which cannot easily converge to the value-oriented methods. It requires hard empirical modelling to endpoints, making it very selective; it cannot have a normalisation step linking to environmental problems as defined at a social and political level; and it has an evaluation method which does not take into account collective aspects as on societal risk aversion. As long as these different paradigms are present, standardisation of impact assessment methods will remain difficult and the evaluation cannot be standardised, not even its structure.

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